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The monitoring of opportunistic macroalgal blooms for the water framework directive

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Abstract

Among the various quality elements which the Water Framework Directive requires should be monitored are macroalgae. One aspect of these is the presence, in transitional waters particularly, of large blooms of opportunistic macroalgae, such as *Ulva* and *Enteromorpha*. Within the United Kingdom (UK) and Republic of Ireland (RoI) there are currently no set ecological quality objectives or standards for macroalgae. Nor are there standard methods for monitoring macroalgal blooms, although various combinations of aerial photography, remote sensing and measurements on the ground are used. This paper attempts to set a logical framework for the prioritisation of sites for monitoring, the development of a tiered monitoring procedure and the derivation of thresholds for classification. Draft threshold limits for percentage cover and biomass of macroalgae have been derived from the literature. The importance of secondary effects and physico-chemical parameters is discussed.

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1. Introduction

1.1. Background

The Water Framework Directive (WFD, 2000) brings a new approach to the regulation and monitoring of controlled waters within the member states of the European Union. Focussing on the overall ecology and function of ecosystems, it brings a holistic approach to the management of rivers, lakes, transitional and coastal waters. Significant pressures on water bodies must be identified and quantified, with confidence ascertained in the classification of the water bodies and the degree of risk to them. Water bodies at risk of failing to achieve Good or High ecological quality status must be assessed, and any necessary remedial measures identified and enacted through a programme of measures. Pressures include habitat loss, hazardous chemi-

* Corresponding author. *E-mail address:* clare.scanlan@sepa.org.uk (C.M. Scanlan). cals and eutrophication. Macroalgae form one biological quality element, with different aspects of these elements being measured in coastal and transitional water bodies.

1.2. Aims

Within the United Kingdom (UK) and Republic of Ireland (RoI) separate monitoring tools are being developed for macroalgae on rocky shores and on soft shores, as the community structure, impacts and physical environment are different in nature. In the UK and RoI, blooms of macroalgae are generally considered to be problems of relatively sheltered, sedimentary shores rather than of hard substrata. Blooms of diatoms and euglenoids can occur (and may confuse interpretation of remotely sensed imagery), but macroalgal opportunists are the main benthic species implicated in nuisance blooms worldwide, and so may be considered to be more significant. Additionally the WFD specifies *macroalgae*. The authors aim to describe the underlying philosophy of the approach taken to the

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development of a tool for monitoring mats of various bloom-forming algae on sedimentary shores, a phenomenon primarily of transitional waters and sheltered coastal areas, and of tentative values for WFD (Water Framework Directive) class boundaries. The main pressure considered is eutrophication.

1.3. Algal blooms and their effects

Macroalgae are natural components of shallow-water marine and transitional soft-sediment communities (Abbott and Hollenberg, 1976). However, excessive growth of opportunistic species may occur under certain conditions, altering the natural balance not only of the algal community, but also of associated faunal communities. Such opportunist species are characterised by high rates of mineral nutrient uptake, nutrient saturation and growth (Wallentinus, 1984), and may have enhanced reproductive capability (Hoffmann and Ugarte, 1985). Blooms form principally of species of Enteromorpha, Ulva, Chaetomorpha or Cladophora, though other green, red (e.g., Ceramium, Porphyra) and brown algae (e.g., Ectocarpus, Pilavella) may also reach nuisance proportions (Vogt and Schramm, 1991; Fletcher, 1996a,b). Hayden et al. (2003), using genetic information, reassigned the genus Enteromorpha to the genus Ulva. However, the traditional generic names represent a useful morphological distinction between the two thallus types and, for the purposes of this paper, the genera will be referred to as separate for the sake of clarity and continuity with previous authors.

Blooms are a world-wide phenomenon (e.g., McComb and Humphries, 1992; Reise and Siebert, 1994; Sfriso et al., 1992; Raffaelli, 1998; den Hartog, 1994; Soulsby et al., 1982), and are reviewed comprehensively in Fletcher (1996a). They most often occur in areas of restricted flushing (Lotze et al., 1999), and are often considered to be the result of nutrient enrichment (Ryther and Dunstan, 1971; Kruk-Dowgiallo, 1991; Nienhuis and Schramm, 1996; Wilkes, 2005), with a concomitant shift from long-lived algal species to short-lived opportunists. Opportunists such as Enteromorpha, Ulva, Chaetomorpha and Cladophora are able to out-compete other seaweeds, taking advantage of nutrient inputs (Rosenberg et al., 1990; Pihl et al., 1999) as well as seagrasses and sometimes phytoplankton, though Twilley et al. (1985) noted that macroalgal abundance itself has been shown to decline under high nutrient regimes due to the attenuation of surface irradiance by blooms of phytoplankton and epiphyte growth. It is well recognised that in the early phase of eutrophication there may be an increase in overall productivity, which may or may not be regarded as beneficial, and high biomass may provide a refuge for small fish, crustaceans and gastropods (Pihl et al., 1996; Raffaelli et al., 1998). However, the process could accelerate to create an undesirable imbalance (Raffaelli et al., 1989), with altered species composition (Norkko and Bosdorff, 1996) and overall reduction in diversity (Jones and Pinn, 2006).

The effects of algal mats are various, and summarised by Fletcher (1996a,b) and Raffaelli et al. (1998), though they have been noted by many authors. Effects include blanketing of the surface causing a hostile physico-chemical environment in the underlying sediment (e.g., Raffaelli et al., 1998; Gamenick et al., 1996), sulphide poisoning of infaunal species (Gamenick et al., 1996), anoxic gradient at the water sediment interface (Norkko et al., 2000), effects on birds (Tubbs, 1977; Tubbs and Tubbs, 1983) including changes in the feeding behaviour of waders (Raffaelli et al., 1989), smothering of seagrass beds (den Hartog, 1994), interference with water use activities by rafts of floating, detached weed (Montgomery et al., 1985) and aesthetic effects such as odour nuisance and deposition on sites such as bathing waters (e.g., Montgomery et al., 1985; Jeffrey et al., 1992). Impacts of algal mats on underlying sediment-dwelling fauna are generally considered to be deleterious (e.g., Soulsby et al., 1982; Tubbs and Tubbs, 1983; Hull, 1987, 1988; Rafaelli et al., 1991, 1998; Raffaelli, 2000; Bolam et al., 2000), although Everett (1994), for example, showed varying effects on underlying fauna. Recent work in estuaries of the south coast of England (Rees-Jones, unpublished) has also shown variable effects on benthic fauna, with sometimes large volumes of overlying weed having little evident impact on underlying fauna.

1.4. Factors controlling macroalgal growth

Various authors (e.g., Lowthion et al., 1985; Poole and Raven, 1997; Elliott and de Jonge, 2002; CEFAS, 2004; Rees-Jones, unpublished) have emphasised that factors such as nutrient supply, temperature, turbidity, bed stability, hydrography and the amount and type of substratum suitable for algal growth are important limiting factors where macroalgal blooms are concerned. As opportunistic macroalgae such as Ulva sp. and Enteromorpha sp. cannot exist above the high tide limit nor grow at depths where turbidity levels limit light intensity (Josselyn, 1985), so the standing stock in an estuary must be limited by the total available intertidal area. Given these pre-conditions, the biomass density will primarily be controlled by nutrient concentrations (up to a physically controlled maximum density). CEFAS (2004) found no evidence that factors such as attachment points, grazing and bed biogeochemistry played a significant role in determining macroalgal growth, although at a local level these may play a modifying role. An assessment of whether an estuary is 'at risk' might therefore focus on issues of availability of intertidal area, turbidity, nutrient concentrations and bed stability. Few studies have shown direct links between nutrient loads and algal biomass, one exception being Lyons et al. (1995). They showed a direct correlation between these two factors, mitigated by the presence of fringing saltmarsh, which may have reduced nutrient loading by de-nitrification.

It is clear that the occurrence, persistence and impacts of macroalgal blooms are governed by a number of physical, chemical and biological factors, which may interact in a complex fashion, and are often difficult to characterise and understand fully.

1.5. Correspondence between WFD and other EU Directives/regimes

Various European Union Directives besides WFD consider the assessment of eutrophication, principally the Urban Waste Water Treatment (UWWTD, 1991), Nitrates (Nitrates Directive, 1991) and Habitats (Habitats Directive, 1992) Directives, and also OSPAR (Oslo and Paris Commission). Unlike WFD, the UWWT and Nitrates Directives each define eutrophication in relation to sources, i.e., nitrogen and phosphorous from discharges, and nitrates from agricultural activities only respectively. OSPAR considers not only the area considered by WFD but also trans-boundary transport across maritime areas. Its Strategy to Combat Eutrophication (OSPAR, 2003) aims "to achieve and maintain a healthy marine environment where eutrophication does not occur", and is developing ecological quality objectives for eutrophication as part of a wider framework that is the basis for an ecosystem approach to management of human activities. As well as definitions of eutrophication, the UWWT and Nitrates Directives also set out measures to combat it through designations as Sensitive Areas (SAs) and Nitrate Vulnerable Zones (NVZs). Existing designations under these directives will remain unchanged by WFD independent of the ecological status of the waters bodies concerned. "Sensitive areas" and "NVZs" will become protected areas under Article 6 and Annex IV of the WFD. UWWT and the Nitrates Directives designations may also be a result of non-eutrophication criteria such as high nitrate concentrations in ground and surface waters for the protection of drinking waters.

The possibility of the different definitions of eutrophication, criteria used and different monitoring regimes of the several directives producing differing assessments of an area, and the implications of this for programmes of measures to combat pollution, has been the subject of much discussion. There is not direct "read-across" between the directives, but a comparison of assessment results is presented in Table 1 (CIS, 2005). The "one-out-all-out" approach of the WFD is at odds with that of the other directives and OSPAR, which use a "weight of evidence" approach to assess eutrophication and the overall status of bodies of water, and for targeting control measures. Potentially WFD could class a water body as Moderate based on nutrients alone without evidence for primary or secondary effects (Leaf, 2006) as is required elsewhere. The need to normalise definitions of eutrophication and have monitoring schemes to produce robust assessment satisfying all relevant criteria have been recognised (COAST, 2002; Leaf, 2006; CIS, 2005). Contributing to this have been discussions on definitions of "undesirable disturbance". Tett, 2004 addresses what may constitute "undesirable disturbance", and Andersen et al. (2006) have proposed a re-definition of eutrophication in terms of undesirable disturbance being the status relative to WFD reference conditions.

The Environment Agency for England and Wales (EA, E&W) has developed internal guidance (Wither, 2003) for assessing the risk to Natura 2000 sites for the Habitats Directive, and has set guidelines (see Fig. 1) for triggering appropriate assessments under the Directive in relation to the extent and density of macroalgal blooms. There is also internal guidance on monitoring algal blooms in relation to the UWWTD. The EA guidelines were derived from the outcome of internal agency discussions using monitoring experience and following a DETR (UK Department of the Environment, Trade and the Regions) workshop attended by leading UK experts (DETR, 2001, unpub-

Table 1

Comparison of assessment results under various policies for waters responding to nutrient enrichment (based on the assumption that WFD is the starting point and that the different sources of pollution are relevant)

Ecological status	WFD normative definition	UWWTD Directive	Nitrates Directive	OSPAR
Assessment	of current status			
High	Nearly undisturbed conditions	Non-eutrophic, designation of sensitive area is not required	Non-eutrophic, not a Polluted Water, designation of NVZ is not required	Non- problem area
Good	Slight change in composition, biomass	Non-eutrophic, designation of sensitive area is not required	Non-eutrophic, designation of sensitive area is not required	Non- problem area
Moderate	Moderate change in composition, biomass	Eutrophic or may become eutrophic in the near future, designation as sensitive area is required	Eutrophic or may become eutrophic in the near future, polluted water, designation as NVZ is required	Problem area
Poor	Major change in biological communities	Eutrophic, designation of sensitive area is required	Polluted water, designation as NVZ is required	Problem area
Bad	Severe change in biological communities	Non-eutrophic, designation of sensitive area is not required	Polluted water, designation as NVZ is required	Problem area

% cover of inter- tidal area	Density	Evidence of biological impact ?	Progress to an appropriate assessment?
<5%			no
5-15%	<100 gm m ⁻²		no
5-15%	100-1000 gm m ⁻²	no	no
5-15%	100-1000 gm m ⁻²	yes	yes
5-15%	> 1000 gm m ⁻²		yes
>15%			yes

Fig. 1. Environment agency Habitat's Directive guidance summarised in tabular form.

lished). This workshop derived tentative criteria for reference levels for algal cover and biomass, based largely on expert opinion.

2. Approach

In trying to arrive at a scientifically sound and environmentally relevant monitoring tool and classification system, we have considered the preceding approaches and guidelines. Using published and unpublished literature we have attempted to derive critical threshold values suitable for defining quality status classes. Results from different studies present problems in that they are often expressed differently, do not provide sufficient levels of detail and do not relate directly to the levels of nutrients or eutrophication in the area of study. We have used expert opinion where standard published levels of effect do not exist. It must be emphasised that the threshold values proposed here are not final and must be validated by testing them against a range of data from sites of different levels of impact. As existing data available to us were not all collected in the same format, some are inappropriate, and others require assumptions or extrapolations to be made. The testing phase is continuing as new data are gathered for the purpose. The degree of confidence in data and in subsequent classification is not presented in this paper, as work is currently underway on determining these.

3. Results

3.1. Setting reference conditions

Monitoring tools should be able to discriminate between the five WFD quality classes, measuring anthropogenically induced deviation from reference conditions. The Directive does not define reference conditions, other than to state that they should be established, be type-specific and represent conditions free from anthropogenic influences. By implication they represent the upper end of High quality status, although High status forms part of a quality continuum and as such incorporates very minor deviation from reference conditions. "Good" and "moderate" deviations from reference conditions (Table 2) similarly are defined only qualitatively by the WFD. The features of macroalgal communities that can be used for assessment of ecological quality should include taxonomic composition and abundance (WFD, 2000).

3.1.1. Taxonomic composition

In macroalgal blooms, taxonomic composition is generally limited to one or more of a number of fast-growing, opportunist species, which tend to have high surface to volume ratios allowing efficient use of available light and nutrients (Littler and Littler, 1980). To aid general understanding of any site of study the dominant alga(e) within mats should be identified to at least genus. However, we consider it scientifically inappropriate and impractical to use taxonomic composition of mats as a classification criterion for several reasons: presence alone of any of the potential nuisance species does not imply deterioration in quality, as they are natural members of the coastal and estuarine soft sediment communities (Abbott and Hollenberg, 1976); the number of species in areas of fluctuating salinity is generally reduced and does not provide sufficient discrimination to classify on number of species (Wilkinson and Wood, 2003). This latter point has also been an issue in developing a tool for monitoring rocky shores within

Table 2

WFD definitions for high, good and moderate ecological status in transitional waters

Quality element	High status	Good status	Moderate status
Macroalgae	The composition of macroalgal taxa is consistent with undisturbed conditions There are no detectable changes in macroalgal cover due to anthropogenic activities	There are slight changes in the composition and abundance of macroalgal taxa compared to the type- specific communities. Such changes do not indicate any accelerated growth of phytobenthos or higher forms of plant life resulting in undesirable disturbance to the balance of organisms present in the water body or to the physico-chemical quality of the water	The composition of macroalgal taxa differs moderately from type-specific conditions and is significantly more distorted than at good quality Moderate changes in the average macroalgal abundance are evident and maybe such as to result in an undesirable disturbance to the balance of organisms present in the water body

estuaries, and is discussed in another paper in this volume (Wilkinson et al., submitted).

3.1.2. Disturbance sensitive taxa

As in (i) above, the presence of opportunistic macroalgal species is not a problem per se and is thus not an appropriate measure. Nor is the absence of other macroalgal taxa a reliable, quantifiable criterion. Seagrasses may occupy the same habit as macroalgae, and are sensitive to high nutrient regimes and smothering by algal blooms (e.g., den Hartog, 1994; Lyons et al., 1995; Foden and Brasier, in press), but again, their absence does not necessarily denote poor quality status.

3.1.3. Abundance

This is defined here as a combination of spatial cover (% cover) and biomass, and these two aspects of abundance form the basis of this tool. Many studies have assessed these in a variety of ways, but for the purposes of future WFD monitoring it will be necessary to adopt a standard approach, so that results are broadly comparable.

In 2000 a DETR (UK Department of the Environment, Trade and the Regions) expert workshop (DETR, 2001) discussed the setting of reference levels in relation to eutrophication for a range of parameters including macroalgal blooms. In the absence of accepted, published standards for cover and biomass, they used expert judgement to propose reference levels. These values of <5% for spatial cover and <100 g m⁻² for biomass were later incorporated by the Environment Agency (England and Wales) into guidance on when to conduct appropriate assessments of areas under the Habitats Directive (Wither, 2003). Microphytobenthos, such as diatoms and euglenoids, were considered by the expert group to have potential for setting standards, but none were attempted, probably due to a lack of suitable data.

It is stressed that the values proposed were derived from expert opinion based on extensive practical experience, but were not objectively derived from, nor then tested against, real data. However, in the absence of a range of unambiguous published values of "harmful" levels of cover or biomass, these were taken here as a starting point.

3.2. Defining boundaries

3.2.1. Percentage (%) cover

While nutrient availability may be a major factor in the increasing dominance of opportunistic green algae in shallow coastal environments, their reported physiological responses to a spectrum of light and salinity conditions show that the area of shore covered is often limited only by the availability of suitable substratum on which to grow (e.g., Poole and Raven, 1997). It is therefore important to define the area of intertidal that may be suitable for macro-algal growth, as stressed by various authors (e.g., Lowthion et al., 1985; CEFAS, 2004). Some areas, e.g., channel edges subject to constant scouring, may never be suitable for

algal blooms and may thus be excluded from calculations of area. Based on various published literature, suitable areas are considered to consist of mud, muddy sand, sandy mud, sand, stony mud and mussel beds. Workers on individual sites must determine the available area based on local knowledge; alternatively the intertidal area as delineated on Ordnance Survey maps could be used. Total available intertidal area is thus the total area available for growth when any known unsuitable areas are excluded.

The DETR expert workshop suggested a reference level of <5% cover of opportunist and climax species for high quality sites (DETR, 2001). This figure was derived from various studies (e.g., Lowthion et al., 1985) as one representing relatively unaffected areas. DETR also proposed that >15% cover represented a problem area. The EA (E&W) have used percentage cover bands, e.g., 0%, 1-25%, 26-50%, 51-75%, 76-100%, for UWWTD and Nitrates Directives monitoring. Greater than 25% cover was considered an indicator of potential harm, with >25% cover within affected areas as an indicator of abundance. This meant concern arose when actual substrate cover would be >6.25% (25% of 25% = 6.25%), similar to the 5% reference level proposed by DETR. Wither (2003) developed guidance for assessing whether Habitats Directive sites were at risk, which built on the DETR discussions, and a summary is presented in Fig. 1. The guidance was intended only to identify when appropriate assessments might be necessary, and not to set assessment criteria per se. Nevertheless we think that the approach gives a useful framework. Wither (2003) stated that problem areas often have 60% of the greater area covered, and that using the above example, i.e., 25% of 60% equals 15%, this would match the 15% suggested in DETR for a problem area. Cover would be assessed as follows (from Wither. 2003):

If survey data show the following:

50 ha with 0% cover, 10 ha with 1–25% cover, 10 ha with 26–50% cover, 5 ha with 100% cover.

The actual amount of substrate covered is

$$\begin{array}{l} ((((1+25)/2\times10/100)+((26+50)/2\times10/100)\\ +(100\times5/100))/75)*100\\ =13.5\% \ \text{not} \ 33\%((10+10+5)/(50+10+10+5))/100 \end{array}$$

While very useful in investigations, 25% cover bands were considered to be too broad for setting classification boundary thresholds for WFD. There is considerable latitude in a band ranging from 1% (High status) to 25% (problem area), with implications for confidence in subsequent classification. For this reason we think it desirable to estimate percentage cover as precisely as possible, e.g., to the nearest 5%, rather than in very broad bands which lose sensitivity. The difficulties of doing this accurately in practise are recognised and discussed later under quality control. In line with the DETR/EA approach, we have adopted <5% cover of opportunistic macroalgae as a reference level (equivalent to High quality status) and propose <15% (=5–15%) cover of opportunistic macroalgae as a threshold level for acceptable cover where biomass is also low (see following sections and Fig. 2). The EA has considered >75% cover as seriously affecting an area, and this could possibly form a threshold for Poor/Bad status with 25–75% delineating a Moderate/Poor band, and 15–25% Moderate.

Percentage cover alone will not indicate the level of risk to a water body, and biomass must also be considered. For example, a very thin (low biomass) covering over 75% of a shore might have little impact on underlying sediments and fauna, yet it still represents a significant deviation from reference conditions. We have adopted an approach whereby a combination of spatial cover and biomass are necessary to achieve a classification. This is being tested on a range of data, but it is not possible to report results at the present time.

3.2.2. Biomass

>3000

DETR (2001) suggested a tentative reference level of $<100 \text{ g/m}^2$ wet weight, but stated that $<500 \text{ g/m}^2$ wet weight was acceptable. The former could therefore form a reference threshold for the High/Good quality status boundary. 500 g/m^2 (slight deviation from High status) would then become the lower limit of the Good class, i.e., the Good/Moderate boundary. Moderate quality status requires moderate signs of distortion and significantly

greater deviation from High status to be observed. The presence of $>500 \text{ g/m}^2$ but less than 1000 g/m² would lead to a classification of Good quality status at best (Fig. 2). but would depend upon the percentage of the area covered. Consideration was given to whether figures for biomass of affected areas (i.e., areas with algal mats) only should be used for establishing boundaries, or whether the mean for the overall area should be used. A further alternative was to use maximum biomass figures. This latter was rejected, as it could falsely classify a water body by giving undue weighting to a small, localised blooming problem. Mean biomass in affected areas is important, as these may form discrete areas within a larger water body, but for classification of the water body as a whole it was considered that the figures expressed in Fig. 2 should be mean figures for the whole available intertidal area. While this could under-represent problems in sub-areas, where higher densities could cause impacts on other parts of the biota or on sediments, data from the sub-areas would be available and would prompt investigative monitoring.

DETR (2001) and others (Lowthion et al., 1985; Hull, 1987; Wither, 2003) have identified 1 kg m^{-2} wet weight as a level of biomass at, or above, which significant harmful effects on biota have been, or may be, observed. Mixed effects have been observed at both lower and higher biomasses, presenting a difficulty with establishing a categorical level of effect. Should this volume of algae occur over a significant area of a shore, this would clearly represent a deviation from pristine conditions, regardless of the occurrence of significant secondary impacts on other parts of the

	g m ⁻²	r ² MODERATE		POOR		BAD			
ALGAL BIOMASS	>1000 to 3000 g m ⁻²	GOOD/MODERATE o entrained algae r ² - monitor		MODERATE	MODERATE/POOR entrained algae - monitor	POOR		BAD	
	500 to <1000 g m ⁻²	to 00 GOOD ²²		GOOD// entrai - m	MODERATE ned algae nonitor	MODERATE		POOR	POOR
	100 to <500 g m ⁻²	HIGH	HIGH/GOOD entrained algae - monitor	G	OOD	GOOD no entrained	GOOD/MODERATE entrained algae	MODERATE	MODERATE/POOR entrained algae - monitor
	<100 g m ⁻²	н	ligh			no monitoring	- monitor	GOOD/MODERATE entrained algae - monitor	MODERATE
		=5%</td <td>>5</td> <td>to 15%</td> <td>>15</td> <td>to 25%</td> <td>>25 to 75%</td> <td>>75 to 100%</td>		>5	to 15%	>15	to 25%	>25 to 75%	>75 to 100%
		% COVER							
		Overlite Obstans Alexa Diseases Alexa Deven							
		High Good		Algai Biomass	:				
				100-500 g m⁻²		5-15%			
		Moderat	te	500-1000 g m ⁻²		15-25%			
		Poor		1000-3000 g m ⁻²		25-75%			
		Ded		$>3000 \text{ m}^{-2}$. 750/			

Fig. 2. Decision table for classification according to biomass and spatial cover (%).

biota or on sediments. This does give rise to potential differences in assessment under WFD, UWWTD, etc. (see above), but accords with the WFD philosophy of deviation from reference conditions. Not all studies have shown harmful impacts at or above 1 kg m^{-2} (e.g., Rees-Jones, unpublished) due to local factors, but a number of studies have shown harmful effects at this biomass to support the DETR expert committee's view that this level of biomass is not acceptable. Threshold values for Moderate/Poor and Poor/Bad are still under discussion, but levels are suggested (Fig. 2) and further scrutiny of existing data, as well as the acquisition of new data, are required before these can be determined with appropriate confidence. Proposed class boundaries to date are summarised in Fig. 2.

The proposed threshold values are expressed as wet rather than dry weight per square metre. Wet weight is commonly used measure, and for practical reasons it is much easier and less time-consuming determinand than dry weight. Various studies have shown good correlations between wet and dry weights (Tindall and Morton, 1998; Lawrence et al., 2000; Pye, 2000; Vila et al., 2001).

3.2.3. Boundary conditions

It has not been possible as yet to establish fully the degree of statistical confidence in the boundaries between the classes, particularly for Poor and Bad, though work is in progress. The presence of weed growing within the surface sediment is proposed as an additional boundary condition. Various workers have noted that persistence of algae within sediments provides both a means for over-wintering of algal spores and a source of nutrients within the sediments (e.g., Raffaelli et al., 1998). External inputs of nutrients to southern UK harbours may support the growth of macroalgae at the start of a growing season but phytoplankton would subsequently restrict nutrient supply, and intense recycling of nitrogen within the sediments is a more likely explanation for continued macroalgal growth (Trimmer et al., 2000). If sediments are consistently anoxic, de-nitrification processes will break down and the system may become self-sustaining (Trimmer et al., 2000). CEFAS (2004) also quantified sediment and coastal water nutrient inputs to various southern English estuaries, finding for Langstone Harbour that total bed nitrogen concentration was comparable to inputs from adjacent coastal waters. If other sources were removed, the bed could be exhausted within a timescale of a few years. In the United States, Thiel and Watling (1998), in following long-term effects of small-scale experiments, found that effects on infaunal colonisers were most severe and long-lasting where decaying algal mats finally became incorporated into the sediment. Build-up of weed within sediments therefore implies that blooms can become selfregenerating given the right conditions (Raffaelli et al., 1998). Absence of weed within the sediments therefore lessens the likelihood of bloom persistence, while its presence gives greater opportunity for nutrient exchange with sediments. CEFAS (2004) found that bed stability was a

determining factor in whether macroalgal blooms had detrimental impacts on underlying sediments and fauna. It is beyond the scope of a macroalgal tool to incorporate physical factors which may mitigate impacts on fauna, whose condition must also be assessed separately. Incorporation of algae into sediments could be considered as a surrogate for bed stability, and indicate the likelihood of blooms becoming self-regenerating. Additionally, incorporation of algal biomass into the sediments in High/Good status water bodies might lead to an increase in the sediment nutrient pool, which could potentially lead to deterioration within class, contrary to the aims of the Directive. Consequently it is proposed that monitoring of Good status water bodies should continue where such a risk is perceived.

3.3. Combining and presenting data

Various approaches as to how cover and biomass values might be combined were considered in establishing boundary values for classes. The proposed approach is to combine % cover with biomass (as wet weight per square metre) in order to obtain a classification (see Fig. 2). Data can be collected for individual affected patches, and give an indication of impact in these areas, but then be applied to the whole available area to represent overall status of the water body. Initial difficulties were seen with the relative size of water bodies. Very large ones could have comparatively low percentage cover overall, yet those patches could still cover many hectares of intertidal. Early consideration of intercalibration data (Wells, personal communication) has highlighted that not taking overall water body size into account would lead to over-estimation of quality status. To account for this, we propose that areal coverage (in hectares) should lower the class of a water body, as derived from Table 2, by one or more classes dependent on the total area of algal mats. This additional layer is outlined in Table 3. The areas expressed here are as yet tentative and need to be refined following testing and intercalibration.

Inter-annual variation in spatial coverage and biomass is well documented and to be expected (Soulsby et al., 1978; Lowthion et al., 1985; Raffaelli et al., 1989, 1999). Though the controlling factors are not always clear, Lowthion et al. (1985), Jeffrey et al. (1992) and Raffaelli et al. (1999) have all drawn attention to the importance of climatic factors in influencing the magnitude of blooms from

 Table 3

 Effect of total patch size on water body class

Total number of hectares of macroalgae	Effect on quality class
<100 ha	No change
100–499 ha	No change
500–999 ha	Downgrade by 1 class
1000–2499 ha	Downgrade by 2 classes
>2500 ha	Downgrade by 3 classes

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year to year. A single year could in theory give a misleading picture if there had been an unusually low growth of algae for any reason. Conversely, an individual year with unusually high values should be investigated, as these may indicate a developing problem. To account for inter-annual variation, it may be appropriate to use rolling means to assess a water body over a period of time, e.g., a WFD reporting cycle. Initially where insufficient data exist for this, trends in data should be examined using appropriate statistical analyses. This harmonises with the OSPAR approach, although within the UK at least monitoring guidance for the UWWTD recommends three years monitoring data collected over a five year reporting cycle. Rolling means should be used in assessing a water body over a reporting cycle. We have still to assess the level of confidence to be represented by annual as opposed to bi-annual sampling.

4. Discussion

The principal methods of assessing spatial cover are listed in Table 4. While we do not recommend one method over another, the pros and cons, as well as costs, should be considered carefully. Estimation of coverage by in situ survey has been shown to give a 10% lower figure than aerial methods, probably due to the different scales used (Wilkes, 2005). In situ measurements may be more accurate than aerial survey methods because of better resolution, but have been estimated as being 10 times slower and more labour-intensive (Berglund et al., 2003 in Wilkes, 2005). With in situ surveys there may be variation between field operators, so suitable training and quality control measures are essential. Raffaelli et al. (1999) found that densities below 1 kg m^{-2} wet weight (a biomass showing impacts on invertebrates in the Ythan estuary) were not visible as clear mats on aerial photographs for the aircraft height and scale used. Remote sensing methods such as CASI are being refined constantly to better discriminate between vegetation types. It is important to determine a suitable degree of ground-truthing for the methodology selected, which may be chosen to perpetuate long-term data sets. In situ estimation of percentage cover can be quality controlled by use of graduated quadrats or the super-position of grids on photographs of quadrats.

Many areas show a spring bloom of algae, followed by a summer dip and then another rise in biomass in late summer. Depending on local patterns it may be necessary to monitor in spring and summer, but peak biomass is most often found in late summer, and is when we would recommend monitoring take place. Rolling of algal mats into ropes can start to occur at this time as weather conditions become more unsettled and, to obtain the most representative estimates of cover, surveys should preferably be carried out before this happens. Ideally mats might be monitored throughout the growing season, but this would be highly resource intensive, and could under-estimate impacts at peak times.

The collection of algal samples for biomass estimation is inherently inexact, as there is no clearly defined point at which all algae may be collected. Some workers have derived biomass data from wet weight surface samples (e.g., Hull, 1987, 1988; Raffaelli et al., 1999), while other workers have taken samples to a depth of one (Pye, 2000) or several centimetres (EA, unpublished). We propose that the surface layer only should be collected, as this is easier to determine once mud is disturbed than one or two centimetres, is likely to be more consistent between workers, and is likely to correlate more closely with what is visible on remotely collected images. Entrainment of algae in sediment has already been acknowledged as significant so, where this occurs, it should trigger appropriate investigation of effects on underlying sediments, e.g., redox profiling, sediment carbon/nitrogen concentrations.

In devising what is in essence a fairly simple monitoring tool, we recognise that the full understanding and characterisation of the causes and effects of algal blooms is

Table 4

Survey methods used for monitoring macroalgal blooms

Method	Comments	Cost
Conventional transects	Application of well understood technique, but labour intensive. Does not provide overall % cover	Low cost, labour intensive
Systematic coverage using hovercraft, etc.	Labour intensive but capable of giving high quality data on both cover and biomass. May conflict with conservation interests	Medium/High
Aerial survey using standard or digital camera	Simple technique; important to ensure adequate resolution is achieved	Low/Medium
Oblique aerial photography	Technique well suited to coverage of smaller locations	Medium
Infra-red false colour (IRFC) stereo pairs	Capable of giving high quality data if undertaken in conjunction with appropriate ground-truthing. Requires expert interpretation	High
Compact Airborne Spectral Imager (CASI)	Some early problems with interpretation, but a method undergoing continuing development	Medium/High, if combined with LiDAR
Satellite, e.g., Quickbird, IKONOS	Good spatial resolution (0.6–4 m); swath width \leq 30 km	Low/Medium
Telescopic surveys	Rapid assessment of sites where access is restricted	Low

Note: All techniques require some degree of ground-truthing. All remote sensing/aerial techniques are dependent on the level of cloud cover for obtaining good results.

actually highly complex, taking in a range of physicochemical, biotic and climatic factors. It was considered unduly complex to develop a tool for one quality element (macroalgae) incorporating criteria for others (invertebrate fauna communities; seagrasses) or groups not even covered by WFD, e.g., birds, and physico-chemical and climatic data. To incorporate all possible factors, e.g., sediment characteristics, bed stability, turbidity, etc., would create a highly complex tool, which would be difficult to apply. While these factors are necessary to gain a comprehensive understanding of the functioning of the system under consideration, they are not necessary to the assessment of whether or not the levels of macroalgae present form a deviation from reference conditions. We have endeavoured to derive environmentally relevant threshold levels of the key criteria of spatial cover and biomass of opportunistic macroalgae using data from published and unpublished literature. This has raised questions where sometimes large volumes of macroalgae have not produced detectable impacts, e.g., on underlying invertebrate infauna, due to mitigating factors locally, but we believe that algal presence must still be considered relative to reference conditions.

Where initial surveys show that further, more intensive work is necessary a programme of investigative monitoring covering all relevant factors, such as substratum stability, nutrients regime, turbidity, etc. should be instituted. Where the possibility exists of deterioration within class, particularly Good status, further monitoring should be undertaken in order to fulfil the "no deterioration" requirement of the Directive. A model developed by CEFAS for the EA (CEFAS, 2004) provides a useful modelling tool for predicting areas of potential growth and the outcomes of varying nutrient inputs.

Work is continuing on developing boundary conditions, and testing the tool against real data. It also continues on preparing standard operating procedures for sampling and analysis for the elements within this tool. Where appropriate data are available, the tool will also be tested in forthcoming intercalibration exercises.

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References

Abbott, I.A., Hollenberg, G.J., 1976. Marine Algae of California Stanford. University Press, Stanford, CA, pp. 827.

- Andersen, J.H., Schluter, L., Aertebjerg, G., 2006. Coastal eutrophication: recent developments in definitions and implications for monitoring strategies. J. Plankton Res. 28 (7), 621–628.
- Berglund, J., Mattila, J., Ronnberg, O., Heikkila, J., Bonsdorff, E., 2003. Seasonal and inter-annual variation in occurrence and biomass of rooted macrophytes and drift algae in shallow bays. Estuar. Coast. Shelf Sci. 56, 1167–1175.
- Bolam, S.G., Fernandez, T.F., Read, P., Raffaelli, D., 2000. Effects of macroalgal mats on intertidal sandflats: an experimental study. J. Exp. Mar. Biol. Ecol. 249 (1), 123–137.
- CEFAS, 2004. Investigation of factors controlling the presence of macroalgae in some estuaries of South East England. Report to the Environment Agency, Southern Region.
- CIS, 2005. Towards a guidance document on eutrophication assessment in the context of European water policies, Interim document.
- COAST, 2002. Guidance on typology, reference conditions and classification schemes for transitional and coastal waters. CIS Working Group 2.4 (COAST).
- den Hartog, C., 1994. Suffocation of a littoral Zostera bed by Enteromorpha radiata. Aquat. Botany 47, 21–28.
- DETR, 2001. Development of ecological quality objectives with regard to eutrophication. Final report, unpublished.
- Elliott, M., de Jonge, V.N., 2002. The management of nutrients and potential eutrophication in estuaries and other restricted water bodies. Hydrobiologia, 513–524.
- Everett, R.A., 1994. Macroalgae in marine soft sediment communities: effects on benthic faunal assemblages. J. Exp. Mar. Biol. Ecol. 175, 253–274.
- Fletcher, R.L., 1996a. The occurrence of "green tides" a review. In: Schramm, W., Nienhuis, P.H. (Eds.), Marine Benthic Vegetation: Recent Changes and the Effects of Eutrophication. Springer, Berlin, Heidelberg, New York, pp. 7–43.
- Fletcher, R.L., 1996b. The British Isles. In: Schramm, W., Nienhuis, P.H. (Eds.), Marine Benthic Vegetation: Recent Changes and the Effects of Eutrophication. Springer, Berlin, Heidelberg, New York, pp. 150– 223.
- Foden, J., Brazier, P., in preparation. Monitoring of seagrasses for the Water Framework Directive.
- Gamenick, I., Jahn, A., Vopel, K., Giere, O., 1996. Hypoxia and sulphide as structuring factors in a macrozoobenthic community on the Baltic Sea shore: colonisation studies and tolerance studies. Mar. Ecol. Prog. Ser. 144, 75–85.
- Habitats Directive, 1992. Directive 92/43/EEC on the Conservation of Natural Habitats and Wild Fauna and Flora.
- Hayden, H.S., Blomster, J., Maggs, C.A., Silva, P.C., Stanhope, M.J., Waaland, J.R., 2003. Linnaeus was right all along: *Enteromorpha* and *Ulva* are not distinct genera. Eur. J. Phycol. 38, 277–293.
- Hoffmann, A.J., Ugarte, R., 1985. The arrival of propagules of marine macroalgae in the intertidal zone. J. Exp. Mar. Biol. Ecol. 92 (1), 83– 95.
- Hull, S., 1987. Macroalgal mats and species abundance: a field experiment. Estuar. Coast. Shelf Sci. 25, 519–532.
- Hull, S., 1988. The growth of macroalgal mats on the Ythan estuary, with respect to their effects on invertebrate abundance. Ph.D. Thesis, Aberdeen University.
- Jeffrey, D.W., Madden, B., Rafferty, B., Dwyer, R., Wilson, J., Allott, N., 1992. Dublin Bay – water quality management plan. Technical Report 7, Algal growths and foreshore quality. Environmental Research Uit, Dublin, 168pp.
- Jones, M., Pinn, E., 2006. The impact of a macroalgal mat on benthic biodiversity in Poole Harbour. Mar. Pollut. Bull. 53, 63–71.
- Josselyn, M., 1985. Do nutrients or physical factors control macroalgal growth in temperate estuaries? Estuaries 8 (2B), 304.
- Kruk-Dowgiallo, L., 1991. Long-term changes in the structure of underwater meadows of the Puck lagoon. Acta Ichthyol. Piscator 22 (Suppl.), 77–84.
- Lawrence, J.E., Grant, J., Quilliam, M.A., Bauder, A.G., Cembella, A.D., 2000. Colonisation and growth of the toxic dinoflagellate *Prorocen*-

trum lima and associated fouling macroalgae on mussels in suspended culture. Mar. Ecol. Progr. Ser. 201, 147–154.

- Leaf, S., 2006. Discussion paper for UK Eutrophication Steering Group and Supplement to UKTAG/POMTT "Use of new standards" paper. Derivation and application of WFD nutrients thresholds, unpublished.
- Littler, M.M., Littler, D.S., 1980. The evolution of thallus form and survival strategies in benthic marine macroalgae: field and laboratory tests of a functional form model. Americam Naturalist 116, 25–44.
- Lotze, H.K., Schramm, W., Schories, D., Worm, B., 1999. Control of macroalgal blooms at early developmental stages: *Pilayella littoralis* versus *Enteromorpha* spp. Oecologia 119, 46–54.
- Lowthion, D., Soulsby, P.G., Houston, M.C.M., 1985. Investigation of a eutriphic tidal basin: Part 1 – Factors affecting the distribution and biomass of algae. Mar. Environ. Res. 15, 263–284.
- Lyons, J., Ahern, J., McClelland, J., Valiela, I., 1995. Macrophyte abundance in Waquoit Bay estuaries subject to different nutrients loads and the potential role of fringing saltmarsh in groundwater nitrogen interception. Biol. Bull. 189, 255–256.
- McComb, A.J., Humphries, R., 1992. Loss of nutrients from catchments and their ecological impacts in the Peel-Harvey estuarine system, Western Australia. Estuaries 15, 529–537.
- Montgomery, H.A.C., Soulsby, P.G., Hart, I.C., Wright, S.L., 1985. Investigation of a eutrophic tidal basin. Part 2 Nutrients and environmental aspects. Mar. Environ. Res. 15, 285–302.
- Nienhuis, P.H., Schramm, W. (Eds.), 1996. Marine Benthic Vegetation. Springer, Berlin.
- Nitrates Directive, 1991. Directive 91/676/EEC on nitrates from agricultural sources.
- Norkko, J., Bonsdorff, E., 1996. Population responses of coastal zoobenthos to stress induced by drifting algal mats. Mar. Ecol. Prog. Ser. 140, 141–151.
- Norkko, J., Bonsdorff, E., Norkko, A., 2000. Drifting algal mats as an alternative habitat for benthic invertebrates: species specific responses to a transient resource. J. Exp. Mar. Biol. Ecol. 248, 79–104.
- OSPAR, 2003. Strategies of the OSPAR Commission for the Protection of the Marine Environment of the North-east Atlantic II Eutrophication.
- Pihl, L., Magnusson, G., Isaksson, I., Wallentinus, I., 1996. Distribution and growth dynamics of ephemeral macroalgae in shallow bays on the Swedish west coast. J. Sea Res. 35, 169–180.
- Pihl, L., Svenson, A., Moksnes, P.-O., Wennhage, H., 1999. Distribution of green algal mats throughout shallow soft bottoms of the Swedish Skagerrak archipelago in relation to nutrient sources and wave exposure. J. Sea Res. 35, 169–180.
- Poole, L., Raven, J., 1997. The biology of *Enteromorpha*. Prog. Phycol. Res. 12, 1–147.
- Pye, K., 2000. The effects of eutrophication on the marine benthic flora of Langstone Harbour, south coast of England. Ph.D. Thesis, unpublished. University of Portsmouth.
- Raffaelli, D., 1998. The community ecology of the Ythan estuary.
- Raffaelli, D., 2000. Interactions between macroalgal mats and invertebrates on the Ythan estuary. Helgol. Mar. Res. 54 (2–3), 71–79.
- Raffaelli, D., Hull, S., Milne, H., 1989. Long-term changes in nutrients, weedmats and shore birds in an estuarine system. Cah. Biol. Mar. 30, 259–270.
- Rafaelli, D.G., Limia, J., Hull, S., Pont, S., 1991. Interactions between the amphipod *Corophium volutator* and macroalgal mats on estuarine mudflats. JMBA (UK) 71, 899–908.
- Raffaelli, D.G., Raven, J.A., Poole, L.J., 1998. Ecological impact of green macroalgal blooms. Oceanogr. Mar. Biol.: Ann. Rev. 36, 97–125.
- Raffaelli, D.G., Balls, P., Way, S., Patterson, I.J., Hohmann, S., Corp, N., 1999. Major long-term changes in the ecology of the Ythan estuary, Scotland: How important are physical factors? Aquat. Conserv.: Mar. Freshwater Ecosyst. 9, 219–236.
- Rees-Jones, E.A., unpublished. Investigation into the impact of macroalgae on intertidal macrobenthos in Kent estuaries. Habitats Directive Stage 3 Review of Consents, Draft Technical Report.
- Reise, K., Siebert, I., 1994. Mass occurrence of green algae in the German Wadden Sea. Ger. J. Hydrogr. 1 (Suppl.), 171–180.

- Rosenberg, R., Elmgren, R., Fleischer, S., Jonsson, P., Persso, G., Dahlin, H., 1990. Marine eutrophication case studies in Sweden. Ambio 19, 102–108.
- Ryther, J.H., Dunstan, W.M., 1971. Nitrogen, phosphorous and eutrophication in the coastal marine environment. Science 171, 1008–1013.
- Sfriso, A., Pavoni, B., Marcomini, A., Orio, A.A., 1992. Macroalgal nutrient cycles and pollutants in the Lagoon of Venice. Estuaries 15 (4), 517–528.
- Soulsby, P.G., Lowthion, D., Houston, M., 1978. Observations on the effects of sewage discharged into a tidal harbour. Mar. Pollut. Bull. 9, 242–245.
- Soulsby, P.G., Lowthion, D., Houston, M., 1982. Effects of macroalgal mats on the ecology of intertidal mudflats. Mar. Pollut. Bull. 13, 162– 166.
- Tett, P., 2004. Understanding of undesirable disturbance in the context of eutrophication, and development of UK assessment methodology for coastal and marine waters. Stage 1 report. What is undesirable disturbance? Report for DEFRA.
- Thiel, M., Watling, L., 1998. Effects of green algal mats on infaunal colonisation of a New England mud flat long-lasting but highly localised effects. Hydrobiologia 375–376, 177–198.
- Tindall, D.R., Morton, S.L., 1998. Community dynamics and physiology of epiphytic/benthic dinoflagellates associated with ciguarera. In: Anderson, D.M., Cembella, A.D., Hallegraeff, G.M. (Eds.), Physiological Ecology of Harmful Algal Blooms, NATO ASI Series G41. Springer-Verlag, Berlin, pp. 293–313.
- Trimmer, M., Nedwell, D.B., Sivyer, D.B., Malcolm, S.J., 2000. Seasonal organic mineralization and denitrification in intertidal sediments and their relationship to the abundance of *Enteromorpha* sp and *Ulva* sp. Mar. Ecol. Progr. Ser. 203, 67–80.
- Tubbs, C.R., 1977. Wildfowl and waders in Langstone Harbour. Brit. Birds 70, 177–199.
- Tubbs, C.R., Tubbs, H.M., 1983. Macroalgal mats in Langstone harbour, Hampshire, England. Mar. Pollut. Bull. 14, 148–149.
- Twilley, R.R., Kemp, M.W., Staver, K.W., Stevenson, J.C., Boynton, W.R., 1985. Nutrient enrichment of estuarine submersed vascular plant communities. 1. Algal growth and effects on production of plants and associated communities. Mar. Ecol. Prog. Ser. 23, 171– 191.
- Urban Waste Water Treatment Directive, 1991. Directive 91/271/EEC, amended 1998 as Directive 98/15/EC.
- Vila, M., Garcés, E., Masó, M., 2001. Potentially toxic epiphytic dinoflagellate assemblages on macroalgae in the NW Mediterranean. Aquat. Microb. Ecol. 26, 51–60.
- Vogt, H., Schramm, W., 1991. Conspicuous decline of Fucus in Kiel Bay (western Baltic): what are the causes? Mar. Ecol. Progr. Ser. 69, 189– 194.
- Wallentinus, I., 1984. Comparisons of nutrient uptake rates for Baltic macroalgae with different thallus morphologies. Mar. Biol. 80, 215– 225.
- Water Framework Directive, 2000. Directive 2000/60/EC of the European Parliament and the Council of 23rd October 2000 establishing a framework for Community action in the field of water policy.
- Wilkes, R., 2005. A desk and field study of biomass estimation techniques for monitoring green tides in the Irish environment. A report to the Irish Environmental Protection Agency, Environmental Research, Technological Development & Innovation Programme.
- Wilkinson, M., Wood, P., 2003. Type-specific reference conditions for macroalgae and angiosperms in Scottish transitional and coastal waters. Final report to Scottish Environment Protection Agency. SEPA Project Reference 230/4136, Heriot-Watt University, Edinburgh, 105pp.
- Wilkinson, M., Wood, P., Wells, E., Scanlan, C, (submitted for publication). Using attached macroalgae to assess ecological status of British estuaries for the European Water Framework Directive.
- Wither, A., 2003. Guidance for sites potentially impacted by algal mats (green seaweed). EC Habitats Directive Technical Advisory Group report WQTAG07c.